Introduction

Marine pollution is a large environmental problem in many parts of the world, and affects both developed and developing countries. Different degradable and non-degradable emissions threaten marine resources and marine biodiversity and, ultimately, the health of marine ecosystems (GPA, 1995; Elmgren, 2001). Large overloads of nutrients are one of many stresses on coastal and marine ecosystems (Gabric & Bell, 1993; Turner *et al.*, 1999; Rabalais, 2000). The growing world population and the associated intensification of agricultural production has led to nutrient enrichment in marine waters over the last decades (Forsberg, 1994; Turner *et al.*, 1999). The spatial distribution of the world population also influences nutrient loads; for example, urbanization has led to a larger outflow of nutrients from ecosystems on land to the aquatic environment. Furthermore, the bulk of the world's population lives in coastal areas, and there is a continuing trend towards its concentration in these regions, which further explains the pressure on coastal and marine ecosystems (GPA, 1995).

It is very likely that nutrient loads will keep on increasing over the next decades. According to the Food and Agriculture Organization of the United Nations (FAO), the world population is expected to rise by 30% until 2030, and this increase will occur almost exclusively in urban areas. The increase in world population makes higher food production necessary. Historically, higher food production has been achieved though rapid intensification of agricultural production; for example, nitrogen fertilizer consumption has increased eight-fold in the last four decades, while per hectare yields of wheat and rice have doubled. Unless substantial action is taken, environmental damage caused by nutrient over-enrichment is therefore likely to remain or increase.

The purpose of this thesis is to analyze how the characteristics of large-scale marine pollution problems affect policy decisions. Marine pollution is usually characterized by multiple, heterogeneous emission sources located in different countries. The properties of the marine recipient vary between different basins, and between coastal areas and the open sea. Multiple, and sometimes interactive, pollutants contribute to degradation of the seas' ecosystems. I investigate how these attributes affect the total costs of abatement and the allocation of abatement efforts between sources, countries and pollutants. There is also considerable uncertainty about natural and economic processes that influence marine ecosystems. I analyze how uncertainty about pollutant transports and abatement costs affects decisions regarding total abatement and the allocation of abatement between countries. The four empirical applications are made to the Baltic Sea, which a semi-enclosed, regional sea in Northern Europe, severely degraded by eutrophication. However, the concepts used are likely to be relevant also for the management of other regional seas.

The thesis is organized as follows; first, a brief introduction to marine pollution problems in general is given. In the next chapter, the economics of marine pollution is discussed and the literature on water management is reviewed. A chapter that presents the main findings in the thesis follows this. Finally, the conclusions are presented.

Sources and environmental effects of marine pollution

Excessive nutrient loads to coastal waters lead to eutrophication. The symptoms of eutrophication are several: increased turbidity of waters; harmful and even toxic algae blooms; oxygen deficiency in bottom sediments; and changes in biodiversity and hence in the very basic conditions for the marine ecosystems (Gabric & Bell, 1993). Scientific studies have shown that the economic value of reduced eutrophication of marine waters can be large (Söderqvist, 1996; Markovska and Zylicz, 1999).

Much of the nutrient pollution that reaches coastal waters originates from economic activities taking place at a considerable distance from the coast, either within the drainage basin or outside it (Turner *et al.*, 1999). The emissions come from agricultural sources, households and municipal and industrial point sources and from combustion of fossil fuels. Many of these sources are non-point sources, where the link between a given emission source and loads to the sea is difficult or even impossible to establish. Most of the emissions are transported from the sources to coastal waters via rivers in the drainage basin, with ocean currents and through the atmosphere, while smaller amounts are deposited directly into the recipient.

Eutrophication of marine waters is linked in several ways to other environmental and resource issues. The pollutants that cause eutrophication contribute also to other environmental problems. In many parts of the world, high nutrient loads from agricultural areas are associated with high levels of nitrate in groundwater (Lundqvist & Falkenmark, 2000). Moreover, nitrogen oxide emissions from combustion of fossil fuels increase the atmospheric deposition of nitrogen on land and seas and, thereby, contribute to both eutrophication and acidification of lakes and soils. Nutrients that reach coastal waters affect marine resources, such as fish stocks. The direction of the impact depends on the characteristics of each species and the interdependencies between different fish populations (Lee, Jones & Jones, 1991; Elmgren, 2001). When observations have shown that fish stocks decline, long-standing controversies regarding the relative importance of pollution and over-exploitation for the decline has slowed down action, even when the situation has become critical. Other environmental problems, such as global warming, can indirectly affect eutrophication processes as these processes depend on both climate and weather (Elmgren, 2001).

Characteristics of marine pollution problems

Large-scale marine pollution is usually associated with complex physical relationships involving several coupled ecosystems, high variability, multiple conflicting objectives and multiple decision-makers (Grigg, 1996). In general, management of water pollution requires an analysis of the different pollutant pathways and the costs of abatement at all different emission sources (Baumol &

Oates, 1988; Segerson, 1988). However, empirical information on pollutant transports and costs is usually hard to find (see *e.g.* Shoemaker, Ervin and Caswell, 1993). When it comes to large-scale marine pollution, the difficulties to obtain good and consistent data are large. Baseline data on *e.g.* emissions from different types of sources, the number of emission sources, and loads and concentrations of pollutants are often uncertain estimates (see *e.g.* Stålnacke, 1996; EEA, 2001). There are complex links between pollutants and abatement technologies. In many cases, several different pollutants can be affected by a single abatement technology (Milon, 1987; Gren, Elofsson and Jannke, 1997). Also, if one abatement measure is introduced, this may influence the efficiency of other abatement technologies, for example if one measure is located upstream while the other is located downstream. The final costs of emission reductions are not known with certainty when policies are decided upon, due to unpredictable technological development, uncertainty about baseline data on the use of technologies and indirect effects on non-polluting sectors.

The dynamics of marine aquifers and their response to changes in pollutant loads are often poorly understood (Turner *et al.*, 1999; Tyrrell, 1999), and difficulties to determine these physical relationships is rather a common characteristic of large-scale marine pollution problems (Gabric & Bell, 1993; Musu, 1997). One reason for the difficulties to quantify these links is that many aquifers consist of several, interdependent basins that differ with regard to their physical and bio-chemical properties. Also, the properties of coastal regions usually differ substantially from those of the open sea. Due to this heterogeneity, the environmental damage caused by pollution varies throughout the water-body. Adding to this, pollutants can interact in different ways during the transport or in the aquifer, and the time for the aquifer to adjust to changes in loads can differ between pollutants (see *e.g.* Stålnacke, 1996; Turner *et al.*, 1999).

Pollutant transports and transformations cannot be predicted with certainty. Both on the way from the sources to the recipient and in the recipient itself, pollutant transports and transformations are stochastic due to variations in weather and other natural processes. Abatement measures can differ with regard to their sensitivity to random processes in the environment. Moreover, if one abatement activity is located upstream, while another is located downstream, this can either increase or reduce the total variability depending on the links between the stochastic processes. When the spatial scale of the pollution problem is larger, the variability associated with a particular abatement measure can differ between regions. Also, the variability of processes in the aquifer itself is a more complex issue to analyze if the aquifer is large than if it is small, because larger aquifers are generally more heterogeneous than are small ones.

The level of spatial aggregation for analysis of large-scale water pollution is not immediately obvious. Policy decision-making does not follow the same geographical boundaries as pollution problems. This implies that policy-makers have to take into account that it is not always possible to differentiate policies between regions with different impact on water quality. When several regions with different policy decision units contribute to the same pollution problem, strategic considerations may affect policy decisions. Further, when policy decisions are taken at national or international level, several different policy areas, with smaller or larger impact on emissions, can be endogenous to the policymaker, and there can be structural impacts on non-polluting sectors from large abatement programs (see *e.g.* Johannesson & Randås, 2000). There are numerous governmental and other agencies at different levels that are involved in policy-making and enforcement (Shortle, 1996; Eckerberg, 1997). Coordination of policies and enforcement across different levels of governments, sectors and regions is therefore a cumbersome task. Often it is not obvious where the ultimate responsibility for pollution control lies. Finally, data on costs for reducing emissions are usually available for administrative regions, such as municipalities, counties and nations, while data on nutrient transports refer to watersheds. Therefore, the choice of spatial aggregation and the number of sectors and decision-makers to include in an analysis requires that a trade-off be made between detail in modeling, consistency of data, and relevance with regard to policy-making.

Similarly as for most environmental problems, the economic value of reduced pollution is not easily quantified. Estimates of people's willingness to pay depend on how researchers describe the effects of pollution reductions to the respondents. Söderqvist's (1998) study suggests that people have very high confidence in the scenarios presented by researcher, and that few respondents doubt the calculations presented. Gaps in scientific knowledge and the role of stochastic processes may not be fully recognized by the respondents in such a study. When several countries are involved, there can be difficulties in comparing studies across countries, and when only few studies are made, it is not self-evident what rule should be applied when extrapolating the results to other countries (*cf.* Söderqvist, 1996; Markovska & Zylicz, 1999). Moreover, seas and coastal regions are used for multiple purposes; for professional and sport fishing, for bathing and recreation and as a pollutant sink. Many people may also consider the ecosystem itself valuable. Estimating these values separately may lead to the wrong conclusions, as the different uses may be complements or substitutes to the users.

As noted above, marine pollution often involves several countries, and it is known from the economic literature that countries that pollute a common natural resource may prefer not to cooperate on abatement, in spite of cooperation being a socially preferred outcome (Mäler, 1989; Barrett, 1994). The reason is that each country can be better off if it abstains from abatement while still being able to benefit from abatement in other countries. Decisions in a single country regarding abatement of emissions to a common sea may also be affected by the difficulties to verify *ex post* whether other countries stick to an abatement agreement, because of the difficulties to measure emissions and loads.

Economics of marine pollution

Economics is not the only scientific discipline that tries to find explanations for and remedies to the degradation of marine ecosystems. However, it contributes to the understanding of the problem by linking choices made by societies, individuals and firms to the corresponding impact on the marine environment.

In this chapter, I first describe economic explanations for pollution. This is followed by a review of literature on water management. At the end of the chapter, I try to identify gaps in knowledge regarding large-scale water pollution.

The theory of externalities and government intervention

The standard economic explanation for pollution is that firms and individuals take their production and consumption decisions without fully considering the effects that these decisions have on others' utility or production (Baumol & Oates, 1988). Pollution is therefore an "externality" that some agents impose on others.

In a situation where if one party makes use of a natural resource, then this affects all other parties, there are reciprocal externalities (see *e.g.* Dasgupta, Mäler & Vercelli, 1997). Reciprocal externalities are typical for common property resources such as seas, lakes, fisheries, the atmosphere, forests and grazing lands. The situation with reciprocal externalities differs from a situation where externalities are unilateral. With unilateral externalities, if one agent makes use of the natural resource, this affects other agents, but the actions of these other agents have no impact on the first agent. One example of a unilateral externality is acid rain caused by people in one country but affecting people in another (Mäler, 1989). Another one is when a country upstream of an international river takes decisions, from which it reaps all the benefits but carry only a small fraction of the costs in terms of pollution or changed water-flow (see *e.g.* Linnerooth-Bayer and Murcott, 1996).

It is well known from economic theory that when there are reciprocal externalities and the users do not cooperate in some way, common property resources are over-exploited (Hardin, 1968). However, both theory and empirical evidence show that non-cooperation and over-exploitation is not inevitable, but that cooperation is possible under many circumstances (Becker & Ostrom, 1995; Ostrom & Schlager, 1996). Costs of monitoring, enforcement and information and preferences for the welfare of future generations are some of the factors that determine whether and how cooperative management develops.

An underlying cause of externalities is the absence of well-defined property rights. Coase (1960) showed that with well-defined property rights, zero bargaining cost, few polluters and few victims, the agents can solve the pollution problem without intervention from governments and the resulting resource allocation will be optimal. However, the major environmental problems today, such as climate change, large-scale water pollution and biodiversity loss, do not fit these simple characteristics. Instead, property rights are ill defined, there are large numbers of pollutants and victims and it is nearly impossible to trace down all affected agents. Bargaining costs would be enormous, and hence markets that solve these problems do not arise spontaneously. Such market failure leads to inefficient resource use.

In the presence of market failure, government intervention can increase welfare and make everyone better off through the introduction of economic incentives that restore an optimal resource allocation. Pigou (1920) showed that an efficient resource allocation could be achieved through the introduction of a so-called Pigouvian tax on pollution. An extensive body of literature explores the use of economic instruments for water pollution and shows that the choice of policy instrument can be a complicated issue when pollutant fate is complex and non-point sources are involved (see *e.g.* Beavis and Walker, 1979, 1983; Segerson, 1988; Shortle, 1990; Malik, Letson & Crutchfield, 1993; Hoag and Hughes-Popp, 1997; Archer and Shogren, 2001).

However, governments do not always act to reduce emissions. Instead, government policies in policy areas other than the environmental have contributed to degradation of marine resources. Subsidies to the agricultural sector have led to intensified crop production that has increased runoff of nutrients and pesticide residues to the aquatic environment in many parts of the world. Also, countries have in many cases failed to agree on policies for fishery, with over-fishing as a consequence. Lack of coordination between the large numbers of different governmental agencies involved in water management is often an obstacle to an efficient allocation of abatement between pollutants, sectors and regions (see *e.g.* Hjorth, 1996; Grigg, 1996; Eckerberg, 1997; Turner *et al.*, 1999).

As noted above, a fully informed, unselfish social planner can, hypothetically, increase welfare through the introduction of suitable economic instruments that reduce pollution. However, in many cases, governments do not even undertake reforms that would benefit all affected agents (Stiglitz, 1998). One reason for such government failure is there can be uncertainty about the direct consequences of the reform. Another reason is that governments cannot commit to a given policy and those affected by the policy know that it can be followed by further reforms in the future. Even if the policy suggested is beneficial to all, there is a risk that the future reforms, to which the policy might lead, can be disadvantageous to certain groups. Moreover, even if there are many reforms that would be beneficial for everyone, such reforms may be complicated. This can be an obstacle, as it requires high efforts from governments and the public to obtain all the knowledge necessary to evaluate such a reform. Instead, simplicity is often required for a reform to be politically appealing. If it is necessary that reforms are both simple and acceptable to all affected agents, then the set of feasible policies may not be very large.

Literature on economics of water management

Empirical literature on the economics of marine management is sparse, while the literature on water quality management in general is abundant. The aim of the following review is to investigate how the spatial scale has influenced studies of water management. The review starts with an attempt to identify a “basic approach” to water management. Then, different extensions to this basic approach are reviewed. Finally, the gaps in the literature are discussed, while comparing the literature to the description of the characteristics of marine pollution in the preceding chapter.

The basic approach

Firstly, it seems to be possible to identify a basic approach in applied analysis of water pollution. This basic approach assumes the existence of a social planner, whose objective it is to either maximize aggregate producer profits or minimize total costs subject to a restriction on the loads of a single pollutant (Braden *et al.* 1989; Johnsen, 1993; Russel & Shogren, 1993; Paaby *et al.*, 1996; Yiridoe & Weersink, 1998). The analysis is static and often only one or two economic sectors are included, typically the agricultural sector and wastewater treatment plants. The research effort associated with these studies is mainly spent on the investigation of abatement costs at different sources and of the different pollutant pathways.

Some of the studies model agricultural crop production and pollutant transports in high detail (*e.g.* Braden *et al.*, 1989; Yiridoe & Weersink, 1998). While modeling crop production in detail, these authors are able to calculate costs from changes in nutrient application, cropping pattern and management measures within a coherent framework. Braden *et al.* (1989) differs from most applied studies by letting the externality – erosion in this case – enter not only the pollution constraint but also the crop production function.

Johnsen (1993) and Paaby *et al.* (1996) extend the scale of the study to the national level. Paaby *et al.* (1996) develop a model of agricultural production, including both crop and livestock production as well as wastewater treatment plants. Johnsen investigates the cost-effectiveness of phosphorus reduction measures in the Norwegian agricultural sector. When extending the analysis to national scale, both Johnsen (1993) and Paaby *et al.* (1996) suggest that abatement costs should be calculated both as private and social costs. An underlying presumption is that other policies than the environmental is endogenous to the policy-maker. Both Johnsen (1993) and Paaby *et al.* (1996) use world market prices for agricultural products in order to calculate social costs. Paaby *et al.* (1996) also adjust the discount rate. The studies illustrate the difficulties associated with the translation of private into social costs in such a model. It is unclear how the social cost approach should be applied consistently throughout a cost-effectiveness study. Not only agricultural markets are regulated, but also for example labor markets, and the opportunity cost of labor is likely to influence cost estimates. Moreover, wastewater treatment plants, included in Paaby *et al.* (1996), are publicly owned, and it is not immediately obvious how the social cost should be calculated, and accordingly, such calculations are avoided by the authors.

Gren, Elofsson & Jannke (1997) and van der Veeren & Tol (2001) extend the analysis of water pollution management to a larger international watershed. Gren, Elofsson & Jannke (1997) analyze cost-effective reductions in nitrogen and phosphorus loads to the Baltic Sea. Both the agricultural sector, wastewater treatment plants, stationary combustion plants and the transport sector are included. Notably, Gren, Elofsson & Jannke (1997) differs from other studies in this review by including not only pollutant transports in water but also airborne emissions. Van der Veeren & Tol (2001) make similar calculations for waterborne nitrogen loads from the agricultural sector and wastewater treatment sector to the river Rhine.

The major advantage of the small-scale studies is that they permit tailoring empirical models to local conditions. For empirical large-scale studies, detail in the modeling of technological relationships is usually sacrificed due to difficulties to obtain consistent data and the need to aggregate data over large regions in order to keep the model tractable. However, if environmental damage is caused by large-scale pollution, then extending the scale of the study also has advantages. Cost-effectiveness then requires that all polluting countries and sources are evaluated simultaneously and that inter-regional pollutant transports are included. This can only be done when including all emission sources that contribute to environmental damage. One can also note that Johnsen (1993), Paaby *et al.* (1996), Gren, Elofsson & Jannke (1997) and van der Veeren & Tol (2001) all, more or less directly, derive the need for nutrient reductions from the international agreements for the North Sea and the Baltic Sea. Van der Veeren & Tol (2001) and Paaby *et al.* (1996) both translate these agreements into national targets by assuming that the 50% reductions for the North and Baltic Seas can be translated into 50% reductions for the river Rhine and for Denmark respectively. However, a cost-effective allocation of abatement between regions is likely to imply different reduction targets for different regions.

The basic approach to water management is extended in several directions. There is a relatively large literature that investigates the role of uncertainty and of dynamics in nutrient transports and transformation. A smaller body of literature assesses the role of multiple pollutants, links between pollution and the marine ecosystem and structural effects of policies to reduce pollution.

Uncertainty about pollutant transports

Uncertainty about the relationship between abatement at the emission sources and the corresponding impact on water quality has achieved a good deal of attention in the literature. Under uncertainty, the allocation on abatement between sources cannot be determined from a comparison of expected marginal abatement costs (Shortle, 1990). Also, it is not obvious how the dual, or cost-effectiveness, problem should be formulated, but several researchers suggest that it is useful to formulate the dual as a cost-minimization problem subject to probabilistic constraints (Shortle, 1990; Horan, 2001). Charnes and Cooper (1963) originally developed methods for optimization under probabilistic constraints in the 1960s, and their approach is used for a relatively large number of empirical studies on water pollution policy. For those applications, it is presumed that the decision-maker takes into account not only expected loads but also standard deviation of loads. Thereby, reductions in pollutant loads have to be larger compared to when uncertainty is ignored, in order to create a safety margin. An underlying assumption is, generally, that damages are convex in pollutant loads. The conclusions from these chance-constrained models depend also on assumptions about the distribution of emissions, see *e.g.* Xu, Prato & Zhu (1996) and Gren, Destouni & Tempone (2000). Common to the papers that use chance constraints is that much of the detail in the modeling of agricultural production has been sacrificed.

Several studies investigate how differences in uncertainty between different abatement options affect total costs and the allocation between measures. McSweeney & Shortle (1990) apply a chance-constrained model to nitrogen losses from a representative farm, assuming that mean and standard deviation of nitrogen runoff differ between crops and management systems. A number of papers focus on the optimal allocation between point and non-point source abatement when nonpoint sources emissions are stochastic and point source emissions are not (Shortle, 1990; Letson, 1992; Malik, Letson & Crutchfield, 1993).

The subject of variability of measures is extended by Kim & Hostetler (1991), that include a single abatement measure in a dynamic model with chance constraints. Milon (1987) includes several pollutants; nitrogen, phosphorus and pesticides, and probabilistic constraints on both surface and groundwater loads. Similarly to McSweeney & Shortle (1990), Kim & Hostetler (1991) and Milon (1987) both assume that nitrogen runoff depends on the choice of crops. Byström, Andersson & Gren (2000) analyze the implication of having one upstream and one downstream abatement measure, when the stochastic behavior of these measures is interdependent. The upstream measure is the choice of agricultural crops, while the downstream measure is wetland creation.

Other studies take into account the spatial aspect of the problem and allow for differences in stochastic processes both between abatement measures and regions. Gren *et al.* (2000a) include both upstream and downstream measures and loads to four different, inter-linked coastal basins with separate load targets. Ellis (1987) incorporates several random parameters and several probabilistic constraints in a purely theoretical model where the choice variable is the BOD removal rate at different river segments and each constraint is a restriction on pollutant load reaching a given section of the river.

Dynamics of nutrient transports and ecosystem response

Several authors analyze the dynamics of pollutant transports and of ecosystem response. Assessments of the role of nutrient dynamics are generally confined to a single pollutant. Including both spatial aspects and dynamics, Fleming & Adams (1997) analyze how the introduction of a fertilizer tax affects ground-water nitrate pollution and Goetz & Zilbermann (2000) investigate theoretically the regulation of phosphorus runoff from agricultural land. Hart (2002) compares the effects of upstream and downstream abatement measures on coastal water pollutant concentrations when emission pathways are complex and subject to time delays.

While disposing of the spatial aspects, Naevdal (2001) and Mäler, Xepapadeas & de Zeeuw (2000) investigate the impact of nutrient loads on water quality in eutrophicated shallow lakes, with a focus on threshold effects in the aquifer. Hart & Brady (2002) compare how different targets for water quality in the Baltic Sea affect the cost-effective abatement path, while including a detailed model of agricultural production and nutrient dynamics, but avoiding spatial disaggregation. In a bioeconomic model of Black Sea fishery, Knowler, Barbier & Strand (2001) show that nutrient load reductions would increase fish catches and lead to substantial economic benefits for fishers, provided that abatement does not lead to a radical shift in the behavior of the ecosystem.

Multiple targets and multiple pollutants

Gren, Elofsson & Jannke (1997) and Milon (1987) both include more than one pollutant, while taking into account joint costs of abatement. The results show that the importance of economies of scope depends on the choice of abatement target. Contrary to those, Connor, Perry & Adams (1995) suggest that there can be diseconomies of scope when there is a need both to control erosion and nutrient runoff from irrigated agricultural production. Milon (1987) and Archer & Shogren (2001) investigate models where pollutants pass through both ground- and surface water and where different constraints are used for those recipients. The papers show that when several receptor points are included, this can have large effects on the optimal allocation of abatement and the choice of policy instrument.

Structural effects of abatement

The economic effects of policies aimed at pollution reduction need not be confined to the polluting sectors, but interdependencies between polluting and non-polluting sectors may spread the impact. Bargur, Davis & Lofting (1971) analyze policies for local pollution control when emissions come from several, interdependent industries and pollutants are non-uniformly dispersing in the recipient. Later studies, where large-scale structural effects are examined, do not include pollutant transports. Johannesson & Randås (2000) develop a multi-country CGE-model for the Baltic Sea region, and conclude that in countries, where the polluting sectors are small relative to the total economy, nutrient load reductions have a small structural impact on non-polluting sectors. However, if the polluting sectors are large compared to the total economy, structural impacts can be large. Wier & Hasler (1999) confirm this conclusion: in Denmark, where the agricultural sector accounts for approximately 5% of total output, increased exports and intensification of agricultural production mainly drive the large increase in nitrogen emissions from Denmark between the 1960s and the 1980s. Structural changes have had minor effects on Danish emissions during the same time period.

Multiple decision-makers

The international aspects of marine water pollution is a little researched area. A paper by Gren (2001) illustrates the difficulties to reach an agreement on international pollution reductions for the Baltic Sea. As suggested by Barrett (1994), a socially better outcome may be achieved if there is a possibility to make international financial transfers. In line with this, Markovska & Zylicz (1999) investigate a cost-sharing scheme for the countries in the Baltic Sea region.

Gaps in knowledge regarding large-scale water pollution

There is a considerable body of literature regarding the economics of small-scale water pollution, in particular as regards the interdependencies between abatement technologies and the spatial, dynamic and stochastic aspects of pollutant transports. Also, multiple receptor points are included in several models.

The methods that have been used for small-scale water pollution have in many cases been applied also for large-scale pollution. There is, however, little work done as regards the dynamics of large-scale pollution, probably due to limited scientific knowledge about the behavior of seas and coastal ecosystems. Few large-scale studies take into account heterogeneity of the aquifer and the physical links between pollutants and marine resources such as fish. Also, few models that include multiple receptor points, such as for example targets for different basins have been found. No models that include targets for both coastal areas and over-polluted groundwater or inland surface water are available.

The role of the complex organizational framework for water pollution policy-making and enforcement is little investigated. For example, the delegation of responsibilities to governments at different levels, and timing of decisions by different authorities have not been examined. In many countries, different interest groups play an important role for the formation and enforcement of policies, but there is little research done regarding the influence that these groups have on policies against marine pollution.

Little research is done that investigates the international aspects of large-scale water pollution and the associated prospects for cooperation or coalition formation. International cooperation regarding climate change and airborne emissions, such as CFCs and acidifying substances, have achieved far more attention in this regard than have water pollution and it is not unlikely that the issues differ in many aspects.

Common to both small- and large-scale water pollution issues is that although uncertainty about nutrient transports is much analyzed, there is little work done regarding the links between different stochastic processes. These links can have a potentially large role for policy decisions regarding both total abatement and the allocation of measures. There is also a lack of knowledge regarding the implications of pollutant interaction and of the links between pollution and environmental damage.

The aim of this thesis is to fill some of the gaps in knowledge regarding management of large-scale water pollution. The particular issues that are treated are described in the next chapter.

Main findings

The purpose of this thesis is to contribute to the knowledge about policies for reducing marine pollution. This is made with the help of economic models with high relevance for empirical policy considerations.

The costs of pollution reductions are a common theme of the thesis. In three of the four included studies, cost-effective reductions in nutrient loads are investigated. These studies show how the minimum total cost is affected by spatial differences in costs and nutrient transports, stochastic nutrient loads, and nutrient interaction. In the fourth study, I analyze whether cost uncertainty provides a rationale for a country to undertake unilateral abatement in order to encourage another country to follow.

The results illustrate how total abatement is affected by the inclusion of two different types of uncertainty; that about nutrient loads and that about abatement costs. Also, they show how the allocation of abatement between measures, regions and nutrients is affected by abatement costs, nutrient transports, nutrient interaction and uncertainty.

All four studies included in the thesis are applied to nutrient emissions to the Baltic Sea, which is a regional sea, located in Northern Europe, and severely degraded by eutrophication. However, the concepts and issues are likely to be relevant also for the management of other regional seas.

This chapter proceeds as follows; first, models, assumptions and limitations in the studies are briefly described; then, the Baltic Sea, to which the studies are applied, is described. This is followed by a discussion of the main findings in each of the four studies included in the thesis. The first article is an analysis of cost-effective reductions in agricultural nitrogen loads to the Baltic Sea, when costs and nutrient transports differ between the regions that surround the Sea. The second extends this analysis to investigate how inter-annual variations in loads affect the cost-effective solution. The third article discusses the role of nutrient interaction and sea currents for a cost-effective abatement program. Finally, the fourth article investigates whether unilateral abatement is beneficial in a situation where it reduces uncertainty about abatement costs.

Models, assumptions and limitations

In articles I-III, a cost-effectiveness approach is used. Cost-effectiveness studies are widely used in empirical research on environmental problems. Generally, for such studies it is assumed that there is an international decision-maker, whose objective it is to reach a given environmental target at minimum cost. One advantage with this approach is that it is possible to evaluate, for a given target, the role of different parameters and processes for the minimum cost and corresponding allocation of abatement, without knowing the economic value of the environmental damages from pollution (Baumol & Oates, 1988). For most environmental problems, these economic values are little known and difficult to quantify. Yet another reason for using exogenous environmental targets in this thesis is that there are international agreements on nutrient load reductions to the Baltic Sea. Article IV differs from the other three articles by modeling the problem as a two-country game, where each country seeks to maximize its own net benefits. The reason for using a utility maximization framework in IV is that the role of cost uncertainty can be analyzed in a convenient way.

Article I provides a basic framework for the remainder of the thesis as regards calculations of costs and nutrient transports. This framework is extended in article II to include stochastic nutrient loads and in article III to allow for nutrient interaction. In article IV cost functions and nutrient transports are highly simplified, although heterogeneity in costs and nutrient transports is still allowed for.

For calculation of the empirical cost functions, it is assumed that prices of fertilizers and agricultural outputs are unaffected by changes in fertilizer use, land use and livestock holdings. For fertilizers, it is reasonable to assume that changes in demand in the Baltic Sea region will not affect the price. Reductions in fertilizer use might however increase prices of agricultural outputs, and reductions in livestock holdings might increase livestock prices. The sensitivity of models with regard to different types of abatement costs has been investigated, and suggests that these effects would not change the substance of the results.

The focus of the thesis is on pollution of marine waters. When reducing loads to coastal waters, this can, however, have effects on inland water quality. Reductions of nitrogen emissions may lead to improved drinking water quality and reductions in phosphorus emissions may reduce eutrophication of lakes. Notably, many of the countries in the Baltic Sea region have unhealthily high nitrate levels in ground and eutrophicated lakes (EEA, 1999a, b). These effects on inland water quality are not taken into account. To incorporate them in the models used in this thesis, it would be necessary include benefit calculations or water quality constraints for receptor points both in the inland and in the sea. Data necessary for doing this are not available.

Also because of data limitations, inland municipal point sources are not included in the studies. The reason is that data on emissions from these sources were not available at the time that articles I and II were written. Later, some data have become available (HELCOM, 2001). Rough calculations that ignore retention suggest that the total emissions from inland sources are about equally large as the total emissions from coastal point sources. Also, airborne nitrogen emissions are excluded from the study. These airborne emissions account for roughly one tenth of the total nitrogen load to the Baltic Sea.

The models used are all static with the exception of article IV, where time is treated in a highly simplified manner. Also, the studies are limited to partial equilibrium analysis. One can note that the results in Johannesson & Randås (2000) indicate that the structural impact of nutrient emission reductions can be large for the countries in transition, and the partial equilibrium approach may therefore imply an underestimation of costs in these countries. Implications of uncertainty for individual decision-making by farmers or wastewater treatment plant operators are not dealt with in the thesis. For a review of how different environmentally related policies affect farmer risk, see *e.g.* Bosch & Pease (2000).

The Baltic Sea

All applications in this thesis are made to nutrient emissions to the Baltic Sea. The Baltic Sea is one of the largest brackish-water seas on Earth, with a fourfold larger drainage basin. Around 85 million people live in the drainage area, and agricultural and industrial production is intensive. Population, in particular the urban population, is heavily concentrated to the coast, while most types of land use are more or less evenly distributed throughout the drainage basin (Turner *et al.*, 1999).

During the last century, nutrient loads to the Baltic Sea increased fourfold for nitrogen and eight times for phosphorus due to increased population and intensification of agriculture and industry within the drainage basin (Gren, Turner & Wulff, 2000). Loads seem to have stabilized in the last decades, at least temporarily. Since the 1980s, riverine loads of nitrogen and phosphorus to the

Baltic Sea have remained fairly constant, although at a high level (Stålnacke *et al.*, 1999).

The physical properties of the Baltic Sea make it particularly vulnerable with regard to pollution. Only certain species can live in the low salinity of the Baltic, and diversity has been further reduced due to the extensive areas of oxygen deficiency (Elmgren, 2001). The excess loads of nutrients lead to frequent and massive blooms of algae. When decaying, the algae cause oxygen depletion in the water and in bottom sediments. Naturally low water exchange between surface and bottom waters contributes to low oxygen concentrations in the deeper parts of the sea. In some years, 25% of the total area has suffered from severe lack of oxygen (Turner *et al.*, 1999). Today, eutrophication is a problem in the entire Baltic Sea, with an exception for the two northern basins, Bothnian Bay and Bothnian Sea. The problems are particularly pressing in certain coastal areas and in the Gulf of Finland and the Gulf of Riga (Gren, Turner & Wulff, 2000).

The Baltic Sea was the first place where environmental changes to an entire marine ecosystem were documented (Gren, Turner & Wulff, 2000). In spite of extensive research on the Sea from the 1960s and onwards there is still large uncertainty regarding

- the role of human activities versus natural processes for the final loads
- the relationship between nutrient loads and concentrations in sea water and
- the different roles of nitrogen and phosphorus for eutrophication.

Some of the main reasons for the difficulty to quantify these relationships is the heterogeneity and variability of the Sea and its ecosystems (Turner *et al.*, 1999; Elmgren, 2001).

The need for future action as regards eutrophication of the Baltic Sea will depend on how the ecosystem responds to measures that have already been undertaken. This response is little known at the moment (Stålnacke *et al.*, 1999; HELCOM, 2001; Elmgren, 2001). The stress on the Baltic Sea ecosystem due to human activities may increase with population increases and with economic expansion in the region. The economies in the former socialist countries are expected to grow, although it is uncertain how long the economic recovery will take. When the economies expand, loads from agriculture and industry are likely to increase (HELCOM, 2001). In the longer term, eutrophication may also be affected by climate change (Elmgren, 2001).

The Baltic Sea Convention

In 1974, the countries around the Baltic Sea signed the Convention on the Protection of the Marine Environment of the Baltic Sea Area. The most important part of the agreement was the establishment of the Helsinki Commission, HELCOM, that should administer the cooperation, promote research, define pollution criteria and adopt recommendations on pollution prevention (Ebbeson, 1996). The 1974 Convention did not stipulate binding measures for land-based pollution, although emissions from land-based sources are of crucial importance for water-quality. The relative weakness of the 1974 Convention as regards the obligations for the countries to pursue pollution reductions can partly be explained by the Convention being signed during the Cold War period. The aim was not only to enhance environmental cooperation in the region, but also to reduce the tension between Eastern and Western Europe (Hjorth, 1996).

The Baltic Convention was reworked in connection with the large political and economic changes in the region caused by collapse of the Soviet regime. A new Convention was signed in 1992, and the new version included more stringent obligations for the countries to reduce marine pollution originating from land-based sources. Still, the new 1992 Convention did not require binding reductions in emissions from the signatories but instead, it stipulated that Best Available Technology or Best Environmental Practice should be used for pollution control (Ebbeson, 1996). Environmental targets, in the form of percentage load reductions, were instead prescribed in the 1988 and 1990 Ministerial Declarations, which do not have the same legal status as a convention. There, it was agreed that nutrient loads to the Baltic Sea should be reduced by 50% between 1987 and 1995. Around 1995, it was clear that this target would not be met and therefore, 2005 was chosen as a new deadline.

The 1992 Convention was an improvement in comparison with the 1974 Convention in that it required regular reports on implementation of the convention by the parties (Ebbeson, 1996). A working document from HELCOM shows that the 50% load reductions were not achieved by far in 1995 (HELCOM, 2001). Load reductions were large in the countries in the transition. This was, however, mainly explained by the economic collapse after the disintegration of the Soviet Union in the early 1990s (Kotov *et al.*, 1997). Industrial and agricultural loads had fallen sharply as a result of the decline in production. Municipal loads had decreased due to considerable investments in the construction and reconstruction of wastewater treatment plants. Denmark, Finland, Germany and Sweden had undertaken relatively smaller reductions than the countries in transition. This can partly be explained by point source load reductions being costly in these countries, as the wastewater cleaning capacity is already high (HELCOM, 2001). Agricultural loads in these countries were little reduced. One possible explanation for this is the strong political role of agricultural interest organizations in these countries (Eckerberg, 1997).

In articles I-III in this thesis, it is assumed that HELCOM has a role as international decision-maker in the region. Accordingly, national policies as well as the Common Agricultural Policy of the European Union, EU, are treated as exogenous. This assumption seems more reasonable for the beginning of the 1990s than for the end. In the early 1990s, the Convention had recently been changed, and when the Soviet Union collapsed, increased cooperation in the Baltic region seemed a plausible future scenario. In the end of the decade however, both Sweden and Finland joined the EU, while Poland and the Baltic States showed their interest to candidate for membership. Although EU has signed the Helsinki Convention, the different roles of HELCOM and EU as regards the responsibilities for management of marine water quality in the Baltic Sea seem not clearly defined. For the future, cooperation in the region is likely to depend on the outcome of the ongoing enlargement process of the EU. Growing trade and increased factor movements between countries may also change the conditions for environmental cooperation (see *e.g.* Mäler, 1989).

Article I – cost-effective reductions in agricultural nitrogen loads

Article I applies the "basic approach", described in the literature review, to riverine loads of nitrogen from the agricultural sector to the Baltic Sea. It includes all countries around the Sea, but not all different sectors that contribute to nitrogen pollution. The study has been used as input in the work of Gren, Elofsson & Jannke (1997).

In article I, it is assumed that an international decision-maker for the Baltic Sea region wants to reduce nitrogen loads from agricultural sources to coastal waters by 50% at minimum cost. The Baltic Sea watershed is divided into 17 regions that differ with regard to abatement costs and nutrient transports. Nitrogen loads can be reduced through changes in land use, improved manure management, reduced consumption of chemical fertilizers and reductions in livestock holdings in all regions. Fertilizer reductions and land use changes are modeled as interdependent with regard to their impact on nutrient loads: if the use of one of these measures is increased, then the impact of the other on nutrient loads decreases. Similarly to other studies applied to large-scale water pollution (Johnsen, 1993; Gren, Elofsson & Jannke, 1997; van der Veeren & Tol, 2001), cost functions for nutrient application, cropping choices and management choices are separable. Data included are, as far as possible, from 1991. In several cases, data on costs and nutrient transports were not available, in particular for the countries in transition. In these cases, data were obtained by extrapolation from other regions.

The results show that if the load target should be reached at minimum cost, Poland has to undertake the largest reductions of all countries, nearly 40% of the total reduction, and carry more than 60% of the total abatement cost. Russia and the Baltic States together account for approximately one tenth of both total abatement and total cost. Abatement in each of the EU-countries; Finland, Sweden, Denmark and Germany, is between 10 and 20% of total abatement, but these countries each carry less than 10% of the total costs.

Two different policy scenarios were compared, the cost-effective solution and a scenario where countries are required to reduce loads uniformly by 50%. This comparison shows the total cost would be nearly 60% higher with uniform reductions. Also, Denmark, Finland and Poland may prefer uniform reductions to Baltic-wide, cost-effective reductions, because their total abatement cost is lower with uniform reductions. This result can be compared with Gren, Elofsson & Jannke (1997), where more sectors and abatement options are covered. There, it is concluded that the costs of uniform reductions are five times higher than when reductions are distributed cost-effectively. Gren, Elofsson & Jannke (1997) also conclude that Poland and the Baltic States prefer a uniform reduction policy to a cost-effective one. Thus, uniform reductions are considerably more costly than are cost-effective reductions, but one obstacle to a Baltic-wide cost-effective abatement program is that Poland may prefer not to cooperate, unless there are institutions that change the distribution of costs to the countries. Markovska & Zylicz (1999) provide one suggestion for the design of such cost-sharing arrangements.

Turning to the different abatement measures, the results from article I suggest that it is necessary to undertake substantial fertilizer reductions if the load target should be reached. Reductions in fertilizer use account for larger reductions in loads than do land use changes. This conclusion coincides with other studies, where large relative load reductions are deemed necessary (*e.g.* Yiridoe & Weersink, 1998; Horan *et al.*, 2001). The results also indicate that manure management can only play a minor role for a Baltic-wide abatement program, due to the small capacity of the measure. Livestock reductions are costly compared to other measures, and play a very small role with regard to cost-effective load reductions. The conclusion that measures in the livestock sector are costly, is consistent with the results in van der Veeren & Tol (2001), although one can note that van der Veeren & Tol (2001) include also other measures at livestock farms, that are less costly than livestock reductions.

Article II – cost-effectiveness with stochastic nutrient loads

Article II expands on article I by taking uncertainty about nutrient loads into account, and by including phosphorus emissions and coastal point sources. It is assumed the international decision-maker's problem is to reduce loads of both nutrients to the Baltic Sea from the included sources by 50% compared to 1997 levels at minimum cost and with relatively high certainty. The model includes probabilistic constraints on nutrient loads to coastal waters. Probabilistic constraints are useful for analysis of uncertainties in the physical relationship between abatement undertaken at different sources and the corresponding environmental damage.

The study includes two types of measures that differ with regard to their stochastic behavior, agricultural measures and measures at coastal point sources. It is assumed that due to weather-driven variations in the retention of nutrients in rivers, the impact of abatement at agricultural sources on coastal loads cannot be determined with certainty, while the impact of abatement at coastal point sources is known with certainty. Moreover, it is assumed that the relative variation in loads from agricultural sources differs between regions and that load from different regions can be correlated. Thus, the policy-maker has to take into account how uncertainty affects decisions regarding both the total level of abatement and the allocation between regions and abatement measures.

Notably, few other empirical studies have analyzed the role of covariance between measures or regions for the allocation of abatement. McSweeny & Shortle (1990) include covariance between measures on farm level, and assume that correlation is strongly positive, but lack empirical data. Byström, Andersson & Gren (2000) investigate the role of correlation between upstream and downstream measures, but have no empirical data on correlation. The novelty of this study is the analysis of the implications of the sign of covariance between regions for costs and allocation of abatement. The empirical model makes use of a variance-covariance matrix of annual nutrient loads, based on data from 1970 to 1993 (Stålnacke, 2000). This matrix shows that for a large watershed such as the Baltic Sea, covariance of loads from different regions can be negative, zero or positive. The sign of covariance is highly related to the meteorological properties of the watershed.

In line with earlier studies, the analysis shows that if the decision-maker cares about load variations then the total cost of load reductions must be higher, compared to the case when variance is ignored, as more abatement must be undertaken in order to create a "safety margin". The empirical results show that if loads to the Baltic Sea should be halved, the total costs can be 80% higher if the target should be reached with 95% certainty, than if the decision-maker ignores variations in loads. The calculations also show that if correlation between regions were ignored, the total costs for reaching the same target would be underestimated by more than 20%.

The theoretical analysis shows that if loads from a particular region are strongly positively correlated with much of the loads from other regions, it is cost-effective to abate more in that region. Thereby, it is possible to keep down the total load in years where these regions have high emission levels. On the other hand, if loads from a region are negatively correlated with much of the loads from other regions, less abatement should be made in that particular region. This is because when loads are high in this region, low loads in many other regions compensate this. The empirical model shows that the inclusion of covariance between regional loads can have large effects on the allocation of costs between different countries, compared to when covariance is ignored.

Turning to a comparison of measures, the results show that for both nitrogen and phosphorus, reductions in point source loads are large in the cost-effective solution. The same holds for nitrogen fertilizers, but reductions in phosphorus fertilizers are less important in a cost-effective program. Land use changes have a larger role for phosphorus than for nitrogen load reductions. The results also suggest that there are economies of scope from coordinated nitrogen and phosphorus reductions if large certainty is required, but that there is little need to coordinate policies for low levels of certainty. This is because for higher levels of certainty, measures that reduce both nitrogen and phosphorus loads, such as changes in land use and reductions of livestock holdings, have to be undertaken if the load targets should be met.

From a policy perspective, it is also interesting to note that taking load uncertainty into account does not change the ranking of the three largest polluter countries with regard to abatement costs, and therefore it is likely to have limited impact on countries position with regard to international negotiations. Also noteworthy, higher certainty requires larger total emission reductions, but it does not change the ranking of different types of abatement measures.

Article III – allocation of nitrogen and phosphorus loads when both nutrients contribute to algae production

The novelty of the study in article III, is that the implications of nutrient interaction for the cost-effective abatement strategy is investigated. Thereby, the study differs from most other applied water pollution studies that are, generally, focussed on a single nutrient. Some researchers analyze nitrogen policies, and motivate this by the health effects of groundwater nitrates (*e.g.* Fleming & Adams, 1997; Yiridoe & Weersink, 1998). Others refer to a politically determined reduction target for the nutrient in question (*e.g.* Johnsen, 1993; Markovska & Zylicz, 1999; Gren, 2001; van der Veeren & Tol, 2001), or by the nutrient being limiting for algae production in the initial situation (*e.g.* Johnsen, 1993; Wier & Hasler, 1999; Goetz & Zilberman, 2000). A few papers include both nitrogen and phosphorus, but have separate targets for the two nutrients, and hence investigate joint costs of abatement but not pollutant interaction (*e.g.* Milon, 1987; Gren, Elofsson & Jannke, 1997).

Thus, the question of nutrient interaction with regard to water quality has achieved little attention in the economic literature, in spite of nitrogen and phosphorus both being necessary for algae growth and hence for eutrophication. This contrasts with the agricultural economics literature, where the substitutability of nutrients with regard to crop production on small plots is treated in a several studies (*e.g.* Ackello-Oguto, Paris & Williams, 1985; Paris, 1992; Chambers & Lichtenberg, 1996). The links between the production function for small plots and the corresponding aggregate production function are analyzed in Berck & Helfand (1990).

According to marine ecology literature, algae take up nutrient in fixed proportions (see *e.g.* Magnusson *et al.*, 1994; Coffaro & Sfriso, 1997). Algae production is to large extent determined by nutrient concentrations in sea water, but the uptake of nutrients from the water differs between algae species, in particular between algae that can us atmospheric nitrogen for their photosynthesis and those that cannot (Tyrrell, 1999). Also, nutrient concentrations vary between different basins of an aquifer and between coastal regions and the open sea. In order to analyze nutrient interaction, it is assumed in this study, that due to heterogeneity of the aquifer and of algae species, the algae production function is continuous and differentiable. Moreover, it is assumed that algae production can change from being nitrogen- to being phosphorus-limited, and that therefore, the algae production function has isoquants that are convex to the origin.

The theoretical analysis shows that if the elasticity of substitution between nutrients is small, emission reductions should, to a larger extent, be focussed on one of the nutrients. The reason is that with a small elasticity of substitution, a reduction in the load of one of the nutrients implies that the other nutrient cannot be used by growing algae and therefore, cannot cause any harm. The nutrient to focus on is not necessarily the one that is limiting for algae production in the initial situation. Instead, the allocation of abatement between the two nutrients is determined by the relative marginal costs of abatement and the relative marginal impact on algae production.

If, on the other hand, costs rise rapidly with the level of abatement and the elasticity of substitution is high, then it can be cost-effective to reduce both nutrients. The reason is that with a high elasticity of substitution, a reduction in one nutrient will not reduce algae production effectively, because the other nutrient will still contribute to algae growth. This situation is more likely to appear when the aquifer is heterogeneous with regard to nutrient concentrations and algae species differ with regard to nutrient uptake from water.

The analysis is applied to cost-effective reductions in algae production in the Baltic Proper, which is the basin in the Baltic Sea that receives the largest nutrient loads. The empirical model indicates that a stronger focus on phosphorus reductions compared to nitrogen reductions can be cost-effective if policy-makers want to reduce algae production. This contrasts with the current Baltic-wide policy with equal reduction rates for nitrogen and phosphorus.

The main explanation for this result is that although nitrogen is currently limiting algae production, this is not due to small nitrogen emissions but to large assimilation of nitrogen emissions by the environment through *e.g.* denitrification. Instead, phosphorus loads are small compared to nitrogen loads. Because of these smaller emissions, it is less costly to abate a large fraction of phosphorus emissions than a large fraction of the nitrogen emissions.

The model in this paper shows the importance of including both nitrogen and phosphorus in economic models of measures against eutrophication, in particular if large load reductions are considered necessary. However, it is important to note that several characteristics of the algae production function are not taken into account in the study. Firstly, the time perspective has not been included. If phosphorus emissions are reduced, it will take approximately 25 years for Baltic Proper to reach a new steady state with lower phosphorus concentrations, while if nitrogen is reduced this will have full effect on nitrogen concentrations in 10 years (Turner *et al.*, 1999). Thus, the benefits of nitrogen reductions can be obtained sooner than can those of phosphorus reductions. Secondly, nitrogen and phosphorus interact in more ways than taken into account in the study, both during the river transports and in the sea (see *e.g.* Stålnacke, 1996; Wulff, 2000). Thirdly, there is considerable uncertainty about the size of phosphorus reductions necessary for the Baltic Proper to shift to phosphorus limitation. Inclusion of these aspects is likely to affect the conclusions as regards the allocation of nitrogen and phosphorus emissions.

Article IV – unilateral abatement under cost uncertainty

Environmental groups sometimes advocate that their own country should undertake unilateral abatement. This is usually motivated with the argument that undertaking unilateral action is at least at step in the right direction, and that if the own country sets a good example, this might affect the behavior of other countries such that they also undertake abatement (Hoel, 1991). One way, by which unilateral abatement could be beneficial, is if it reduces uncertainty. It is commonly suggested in the political debate that unilateral action in the form of pilot projects, or promotion of new and environmentally friendly technologies, should be undertaken because the benefits from providing new information can be high (Schmidt, 1998). Hoel (1991) suggests, that if there are producer lobby groups in a foreign country that argue that abatement costs are high, unilateral action in the home country can be used to demonstrate that such reductions can actually be made without high costs.

Article IV investigates whether uncertainty about the unit costs of abatement can be an incentive for a self-interested country to undertake unilateral action. To my knowledge, there is no other study where the role of cost uncertainty for unilateral abatement is investigated.

Abatement costs for different environmental policies are rarely known when a policy is decided upon. Harrington, Morgenstern & Nelson (2000) analyze how *ex ante* estimates of abatement costs differ from the *ex post* observations. They conclude that the *ex ante* unit costs of abatement can be inaccurate if, for example, technological innovation is not accounted for, cost information is out-of-date, or underestimation of costs can be embarrassing for the persons responsible. Uncertainty about the unit cost of abatement is thus strongly linked to the use of technologies.

Article IV assumes that there are two non-identical, risk-averse countries and that the unit costs of abatement in the two countries are uncertain but correlated. Such correlation can be explained by *e.g.* similarities in abatement technologies. Moreover, it is assumed that the countries take a once-and-for-all decision on a large, irreversible abatement program and that these decisions can be taken either simultaneously or sequentially.

Intuitively, one could imagine that with correlated abatement costs, undertaking unilateral action in a country, say country 1, may reduce uncertainty about the cost in another country, say country 2, if country 2 waits and observes *ex post* abatement costs in country 1. When learning about *ex post* abatement cost in country 1, country 2 is able to make a more precise guess about costs in the own country. When uncertainty is reduced in this way, country 2 might be willing to abate more. Thereby, utility in country 2 could increase due to the reduction in cost uncertainty, and utility in country 1 could increase due to higher abatement in country 2.

However, as this study shows, the above intuitive reasoning ignores important aspects of the problem. As is shown in the article, unilateral abatement would reduce uncertainty about the unit costs of abatement more, if a country with small uncertainty would undertake abatement first, while a country with (initially) large uncertainty would follow. But for reasonable parameter values, it is quite possible that neither country would like to participate in a game where the country with low cost uncertainty moves first. The country with low cost uncertainty might not like to make the first move because when cost uncertainty is large in the follower country, the first mover will be highly uncertain about the follower's future abatement decision. This uncertainty reduces utility in the first country, because this country will not know in advance if it chooses the best abatement level in relation to the level that will actually be chosen by the follower country. The follower country, on the other hand, may not benefit at all from the reduced cost uncertainty. The reason is that the first country can shift over much of the abatement burden on the follower country. When the follower has to undertake more abatement, variance of total cost may be unchanged or even increase, even if variance of the unit cost is reduced.

Instead, both countries may prefer taking their decisions simultaneously, or let the country with high cost uncertainty move first, such that unilateral action will not reduce cost uncertainty to any larger extent. One can note that the conclusion, regarding which country prefers to act as Stackelberg leader, is consistent with the results in Albaek (1990), where uncorrelated cost uncertainty is modeled in a duopoly setting.

A numerical example, where Sweden and Poland decide on nitrogen abatement levels under correlated cost uncertainty, illustrates the model in this study. It is assumed that the two countries have similar abatement technologies and therefore positively correlated unit costs of abatement. Also, it is assumed that Polish abatement costs are lower than Swedish, and that therefore, the variance of Polish abatement costs is smaller. The results suggest that both countries would prefer either to take abatement decisions simultaneously or let Sweden abate first, while Poland follows, compared to letting Poland move first.

The results from the numerical example also suggest that unilateral action under cost uncertainty leads to an increase in total emissions, compared to when countries take their abatement decisions simultaneously. This result is obtained independently of whether the country with high or low cost uncertainty moves first. These results contrast with the results in Hoel (1991), where it is concluded that under non-cooperation, unilateral action leads to smaller total emissions. The differences in results is explained by different interpretations of unilateral action, Hoel (1991) interprets unilateral action as a country acting as if it has larger benefits from abatement than in actually the case, while in this thesis, unilateral action is interpreted as one country acting before another.

Conclusions

This thesis fills some of the gaps in knowledge regarding policy-making for large-scale water pollution. The role of heterogeneity of the watershed and the aquifer for policy-making is analyzed, as well as the role of uncertainty and nutrient interaction. The results show that economic considerations are of large importance for policies that are intended to deal with uncertainties about natural processes and technologies as well as pollutant interaction. Thus, economists can contribute to policy-making through analysis of how these issues are related to costs and benefits of abatement and how they affect international negotiations between countries. In this final chapter, I first summarize the contribution of the articles to the literature. Then, I present policy suggestions with regard to nutrient pollution of the Baltic Sea and suggestions for further research.

Contribution of articles to the literature

There are four main messages that emerge from the articles; the first concerns the relationship between different random events; the second deals with the role of the ecosystem response to emission changes while the third is about the role of low-cost countries with regard to pollution of a common, international resource and the fourth pertains to the scientific value of "simple" models.

Firstly, articles II and IV suggest that when the spatial scale of the analysis is large, random events can be different across regions, but they might be interrelated. These studies show that it matters for the policy conclusions whether uncertainties are treated as interrelated or not. In article II, I show that the fact that pollutant loads from different regions are stochastic and correlated has large impact on total minimum abatement costs and the allocation of costs between countries. In article IV, it is shown that correlated cost uncertainty has considerable implications for countries' decisions on whether they should undertake abatement before other countries, or wait for other countries to move first. The studies show that links between different random events can have large implications for the optimal choice of policy and that policy decisions in general affect the extent of both environmental and economic risk.

Secondly, in articles I-III, I have analyzed the implications of treating aquatic ecosystems not as inert media that passively receive emissions, but as active production systems. In article I, this is made in a simple way, by taking into account nutrient assimilation. This is extended in article II to include variability of nutrient assimilation, and in article III to allow for nutrient interaction. The response of the ecosystem to changes in emissions is shown to be of large importance for policy consideration, both as regards the total extent of action and the choice of emission sources and pollutants to address.

Thirdly, articles I and IV both address the issue of how individual countries may act when confronted with an international pollution problem. In article I, it is concluded that countries with low abatement costs may prefer a uniform reduction policy compared to an international, cost-effective policy, because the uniform reduction policy can be associated with lower total abatement costs for these countries. In article IV, it is suggested that uncertainty of abatement costs might be smaller in low-cost countries than in high-cost countries. It is shown that countries with low cost-uncertainty are likely to prefer to wait for other countries to undertake abatement first, compared to abating prior to others. Thus, the results suggest that there might be double reasons for low-cost countries to be passive with regard to pollution of the international commons. The first reason is that they may want to avoid high costs associated with cost-effective reduction. The second reason is that waiting can reduce environmental risk in these countries.

Fourthly, article I, where costs and pollutant transports are modeled in a relatively simple manner, has proven to be useful as a baseline scenario, with which to compare the other studies. Such simple models can be useful both as a basis for further research, and for measuring the magnitude of the importance of additional aspects of the problem.

Policy suggestions for the Baltic Sea

There are at least two policy issues that could be discussed in light of the results from this thesis. The first issue is whether and how policies should be accommodated to uncertainty and the second is whether there is a need to reform the current *de facto* policy with uniform reduction rates for nutrients and countries (see *e.g.* HELCOM, 2001).

The results from the thesis suggest that the fact that nutrient loads to coastal waters are stochastic and vary between years does not have any impact on the ranking of cost-effective abatement measures or on the ranking of countries with regard to abatement costs. However, the existence of load variations may still provide a reason for policy change if peak loads in single years cause high environmental damage. In that case, it is necessary to undertake more abatement in total, in order to have a safety margin with regard to natural variations in loads.

The thesis does not support unilateral action that is undertaken with a purpose to reduce uncertainty about the costs of abatement, if it is necessary to undertake a large, irreversible abatement program in order to obtain accurate cost information. On the contrary, unilateral action that reduces uncertainty efficiently can lead to larger total emissions and lower utility to the involved countries. One question that arises here is whether large nutrient reductions to the Baltic Sea are necessary to obtain accurate information about the costs for reducing nutrient loads to the Baltic Sea. It seems to me that this might indeed be the case. In spite of many years of abatement projects in different parts of the watershed, there is still considerable uncertainty about the costs associated with the international load reduction targets. One of the reasons for this is the large number of non-point sources contributing to pollution, and the difficulties associated with the calculation of costs associated with policies that reduce emissions from those. Thus the conclusions as regards unilateral abatement might well apply to nutrient reductions to the Baltic Sea.

Turning to the issue of uniform reductions for countries and nutrients, the thesis confirms the old wisdom that there are considerable cost savings to be made from abandoning the approach with uniform reductions (see *e.g.* Tietenberg, 1985). Taking first the current *de facto* policy with uniform reductions for all countries, it is clear from this thesis as well as from other studies that there would be large savings from a reallocation of abatement efforts between countries. In particular, an institutional arrangement that would encourage Polish abatement would be beneficial to the whole region. Secondly, as regards the uniform reduction rates for nitrogen and phosphorus, the results indicate that this policy can be costly compared to a policy that aims at reducing eutrophication. The results suggest that undertaking more phosphorus reductions and less nitrogen reductions might save costs. It is important to note that the conclusions as regards the allocation between nutrients are of a preliminary nature, and that the issue needs further research. If the conclusion holds, however, and more phosphorus reductions should be undertaken, this implies larger costs in the countries in transition, where phosphorus loads are high, because wastewater treatment is less developed.

Thus, if the uniform policies for countries or nutrients are to be abandoned for a cost-effective policy then, in either case, more abatement should be undertaken in the countries in transition. For this to reduce costs in reality, it might be necessary to improve international financing institutions and the coordination between them. Today, the extent and effectiveness of foreign support to water quality improvements in these countries is not well documented, and activities are little coordinated (Hjorth, 1996).

Suggestions for future research

There are at least two interesting extensions possible to articles II and III. Firstly, it would be possible to investigate the role of stochastic loads in a model with nutrient interaction, while making use of available empirical data on variance and covariance of nitrogen and phosphorus loads. This analysis can be carried out using a model with probabilistic constraints, similarly as in article II. Such a study could provide useful knowledge on the role of covariance between nitrogen and phosphorus loads for costs and the allocation of abatement between the two nutrients. Secondly, it would be interesting to carry out an analysis of how uncertainty about the shape of the algae production function affects the allocation between nitrogen and phosphorus reduction. In particular there is uncertainty about the point where the system switches from nitrogen to phosphorus limitation. This analysis would make possible to analyze how the advantage of being able to reduce environmental damage through reductions in a single nutrient, such as discussed in this thesis, is affected by uncertainty about the point where nutrient-limitation shifts.

Article IV can also be extended in several ways. Firstly, one subject would be to investigate theoretically whether there is a link between the endogenous choice by countries to play either a Cournot or a Stackelberg game, and the corresponding total emission level. Compared to the analysis in article IV, this would require a simplification of the payoffs and possibly also of the type of uncertainty. One point of departure for such a model is *e.g.* Haslbeck (1995), where sequential contributions to a public good under asymmetric information is modeled in a Stackelberg game. There are also several models that endogenize the choice of game to be played, Cournot or Stackelberg, in a duopoly setting (Sadanand & Sadanand, 1996; van Damme & Hurkens, 1999)**.** Such as study could give more information about situations when unilateral abatement will or will not lead to more emissions.

A second possible extension of article IV is an analysis of how a social planner should, optimally, choose the sequential order of different abatement projects. Looking at the Baltic Sea region, there are numerous local and regional abatement projects. The choice of location and design for these projects may today be determined by different factors, such as *e.g.* severity of local environmental problems, availability of expertise and attitudes of local governments and interest organizations. The results from these projects are usually reported and documented, and used when planning new abatement projects. For these new projects, the results from earlier projects can be used, not as absolutely accurate predictions of the parameters that will apply for the new project, but as indicators. Parameters are likely to be correlated between projects but not identical. One question here is how a utility maximizing social planner should determine the order and size of projects given his beliefs about how valuable the information from a project will be for other projects. One point of departure for such a study could be a model of adaptive utility (see *e.g.* Cyert and de Groot, 1987).

The role of the complex organizational framework for nonpoint pollution management is not analyzed in this thesis, although it is a field of large importance for reductions of non-point source emissions. One interesting subject for a study is the local enforcement of nationally determined environmental policies. Studies by political scientists (Eckerberg, 1997) suggest that nationally determined policies for non-point pollution are often weakened by inefficient local enforcement. The determinants of environmental performance by governments and agencies has been analyzed by several researchers (Dasgupta *et al.* 1995; Jahn, 1998; Karkkainen, 2002). To my knowledge, there has been no attempt to analyze the local environmental performance with regard to water quality. It could be investigated empirically, to what extent local governments take into account the impact of emissions from wastewater treatment plants on the Baltic Sea. When deciding on the level of wastewater treatment, self-interested municipalities may take into account the impact on the Baltic Sea, but also *e.g.* local environmental effects, the preferences of local voters and the possibility to be re-elected in the future. For several of the countries surrounding the Baltic Sea, there are relatively good data available on wastewater treatment plants, municipalities, regulations, local water quality and the impact on the Baltic Sea.

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